CHAPTER 2

LITERATURE REVIEW

2.1 GENERAL

In wastewater treatment technology, the principles of SBRs operated in a fill-and-draw mode, have been known from the very beginning of the activated sludge process. Since the 70s (Goronszy, 1979), SBRs have become a common modification of activated sludge process, especially for research purposes. Currently the primary area of SBR applications is in research into nutrient removal processes and population dynamics of activated sludge.

The SBR process is an activated sludge process, does not bear any similarity to the extended aeration process. In fact, unlike an EA process the SBR is designed to operate under non-steady state conditions. An SBR operates in a true batch mode. **Figure 1.1,** shown previously, is a schematic presentation of the SBR process. As such, it provides for inherent flow equalization and flow blending, and also reduces operator skill and attention requirements (USEPA, 1986a).

Advantages of SBR operation include:

- Elimination of a secondary clarifier and Return Activated Sludge (RAS) pumping. This is because the SBR tank acts as a clarifier during the Settle phase of the cycle.

During this time, the activated sludge settles to the bottom and the clear supernatant is observed at the top of the tank. The supernatant is decanted away, while the activated sludge remains in the tank to provide MLSS for the next React cycle. Hence, there is no need for secondary clarifier and RAS as in the case of conventional activated sludge systems. (Tchobanoglous and Burton, 1991).

- High tolerance for peak flows and shock loadings. Since the SBR operates in batch mode, an equalization tank or multiple SBR tanks are used to accommodate continuous inflow of wastewater (WEF, 1992). These will also be able to accommodate peak flows and shock loadings without adverse impact on the performance of the plant.
- Avoidance of MLSS "washout" during peak flow events. Since the activated sludge or MLSS is retained in the SBR tank at all times, and only discharged during sludge wasting, there is no opportunity for MLSS washout during peak flow events. Furthermore the flow equalization characteristics of the SBR system described earlier will be able to accommodate peak flows without involving MLSS washout (WEF, 1992).
- Clarification under ideal quiescent conditions. During the Settle phase, there is no aeration, no mixing and no decanting occurring. Thus the SBR tank is in quiescent conditions. This allows for settling or clarification under ideal "no movement" conditions (WEF, 1992).
- Process flexibility to control filamentous bulking. Various researches (Chiesa and Irvine, 1985; Wanner, 1992; Casey *et al.*, 1994) demonstrated experimentally that frequent shifting of activated sludges between feast and famine conditions is an

effective means to control excess growth of filamentous organisms. Many filamentous organisms are aerobic and can be destroyed by prolonged periods of anaerobiosis. Thus anaerobic or anoxic conditions maintained within the process will restrict the growth of these filaments (Eckenfelder, 1989a). Filamentous organisms are discouraged in systems that have high growth rates (i.e. at elevated substrate concentrations) and prolonged starvation periods (Irvine *et al.*, 1997).

The major disadvantage is the relative lack of operational experience to date. Operational and performance reviews of several conventional SBR plants in operation in Canada, Australia and the US in 1984 (WEF, 1992) indicated that no two plants evaluated used the same operating strategy even though their goals were identical, i.e. removal of BOD and SS. Arora *et al.* (1985) reported on eight installations, recording that total cycle times varied from 7 hours to 49 hours, food-to-microorganism ratio ranged from 0.03 to 0.18 per day and sludge age varied from 15 to 80 days. In most cases, effluent quality was both satisfactory and reasonably stable.

Table 2.1 shows the operational and performance summaries of eight conventional SBR plants in operation in Canada and the US in 1984. (Arora *et al.*, 1985; USEPA, 1986a; USEPA, 1987). The designs differ for several components of these plants including inlets, aeration and mixing systems, and decanters, but their operation follows sequencing batch principles. Note that a review of the information contained in **Table 2.1** indicates that no pair of the plants evaluated used the same operating strategy even though their goals were identical (removal of BOD and SS) (WEF, 1992).

Figure 2.1 is a histogram depicting the scatter of daily effluent data for the Culver, Indiana SBR (USEPA, 1983). As indicated by this case history performance summary data, effluent quality was both satisfactory and reasonably stable (USEPA, 1983). The total number of observations was 407, over the period between May 1980 and May 1981. The daily average raw BOD_5 and SS concentrations were 160 mg/l and 130 mg/l respectively. Both daily average BOD_5 and SS concentrations in the effluent (before chlorination) were less than 10 mg/l in both tanks. Post chlorination effluent BOD_5 averaged 5 mg/l (USEPA, 1983).

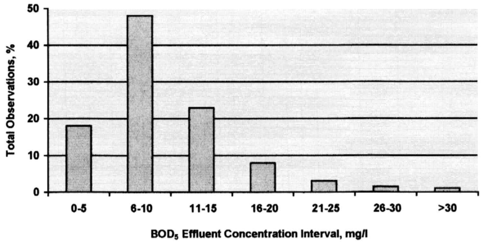

Figure 2.1: BOD_5 Performance Histogram For SBR At Culver, Indiana

Source: USEPA, 1983

Table 2.1: Evaluation Summary for SBRs

Parameter	Canada		USA				Australia	
	Rivercrest, Manitoba	**Glenlea, Manitoba**	**Choctaw, Oklahoma**	**Grundy Center, Iowa**	**Eldora, Iowa**	**Culver, Indiana**	**Tamworth, New South Wales**	**Yamba, New South Wales**
Mode of operation	SBR	SBR	SBR	SBR	SBR	SBR	ICEAS	ICEAS
Date when operation commenced	August 1983	1978	August 1983	June 1983	April 1984	May 1980	June 1983	June 1983
Design average flow, gpd	24,000	2,000	500,000	832,000	220,000	-	535,000	253,000
Design loading								
BOD, mg/l	236	251	366	200	120	170	260	260
SS, mg/l	200	152	350	-	-	150	-	-
NH_3, mg/l	37	55	19	15	25	20	35-40	-
Current average flow, gpd	60,000	1,165	200,000	800,000	220,000	353,000	535,000	-
Desired effluent quality								
BOD, mg/l	TOC-40	30	20	30	30	10	30	30
SS, mg/l	30	30	20	30	30	10	30	30
NH_3, mg/l	-	-	15	6 (summer), 11 (winter	8 (summer), 10 (winter	-	-	-
Actual effluent quality				Not being met because of decanter problems	Data was not available. Effluent appeared to be satisfactory.			
BOD, mg/l	11	5	8			10	5 to 10	6 to 10
SS, mg/l	15	6	18			5	5 to 10	10 to 15
NH_3, mg/l	10	2	-			1.0	2.2	1.0
Mode of operation at design flow								
Fill time	-	-	-	40 minutes (without air/pumps)	-	180 minutes (30% mixed, 70% aerated)	Continuous	Continuous
React time	90 minutes	22 hours	18 hours	120 minutes (with air/pumps)	150 minutes	42 minutes	120-150 minutes	150 minutes
Settle time	45 minutes	1 hour	3 hours	60 minutes	80 minutes	42 minutes	45 minutes	180 minutes
Decant time	20-60 minutes	1 hour	3 hours	40 minutes	50 minutes	42 minutes	45 minutes	45 minutes
Idle time	-	-	-	60 minutes	45 minutes	60 minutes	-	-

Table 2.1: Evaluation Summary for SBRs (continued)

Parameter	Canada		USA				Australia	
	Rivercrest, Manitoba	Glenlea, Manitoba	Choctaw, Oklahoma	Grundy Center, Iowa	Eldora, Iowa	Culver, Indiana	Tamworth, New South Wales	Yamba, New South Wales
Important design parameters								
DT, hours	7.6	49	48	20.4	43	16.5	46	36
F:M, kg BOD/kg MLSS	0.18	0.032	0.037	0.078	0.05	0.08-0.16	0.04	0.05
SRT, days	43	18.8	0.028 Sludge wasted twice in 10 months	0.067	25.3 Sludge not wasted in last 2 months	15.45	-	-
Power usage, kWh/kg BOD applied	0.8	22.9	2.9	0.8 to 1.3	2.2	2.1	1.9	1.5
Unit processes								
Trash rack	Yes	-	Yes (bypass)	Yes (bypass)	-	-	-	Yes
Mech. Screens	-	-	-	Yes	Yes	Yes	Yes	-
Comminutor	-	-	Yes	-	-	Yes	Yes	-
Grit removal	-	-	-	Yes, aerated	Yes, aerated	-	-	Yes
Equalization	Yes	Lift station wet well	Emergency holding pond	Sideline equalization	-	-	-	-
Primary	-	-	-	Yes	-	Yes	-	Yes
Treatment SBR	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Disinfection	-	-	Yes	-	-	Yes	Yes	Yes
Sludge treatment	Holding tank and land application	Agriculture farm	Holding tank and land application	Aerated sludge holding and sludge beds	Anaerobic digesters and sludge beds	Aerobic digesters and sludge beds	Polishing lagoon, Sludge lagoon	Polishing lagoon, Aerobic lagoon
Reasons for providing this technology	Capital cost savings and simple operation	Capital cost savings and simple operation	8.4% savings in life cycle costs	19% capital cost savings in secondary treatment process, or 8% savings in overall plant cost	Capital cost savings and simple operation (100% city funding)	Full-scale study funded by EPA	Capital cost savings	Capital cost savings

Source: WEF, 1992

Table 2.2 shows some case history performance and operating summary data for the Grundy Centre, Iowa, SBR. The data were obtained between June 10 and July 10, 1985. As shown by the table, effluent concentrations averaged less than 8 mg/l for BOD_5 and about 12 mg/l for SS. During the test period, the influent flow rates averaged about 80% of design (USEPA, 1987).

Table 2.2: Performance Data for Grundy Center, Iowa, SBR

Parameter	Influent	SBR with React	SBR without React
Flow, m^3/d [a]	2580	-	-
BOD_5, mg/l	220	8	8
SS, mg/l	170	12	12
NH_4-N, mg/l	17	1.1	1.0
NO_x-N, mg/l	0.7	2.1	2.6
P, mg/l	8.3	3.7	4.9
MLSS, mg/l	-	1700	1800
WS, %	-	1.2	1.2
SVI, mg/l	-	130	160
Approximate detention time for both tanks			25 hours
Approximate organic loading for both tanks			0.12 kg BOD_5/ kg MLSS.d

[a] m^3/d x 0.000264 =mgd

Source: USEPA, 1987.

The SBR can be operated to achieve combined carbon and nitrogen oxidation, nitrogen removal, and phosphorus removal (USEPA, 1988). The process shown in **Figure 2.2** will accomplish all the above three processes (Arora, 1985).

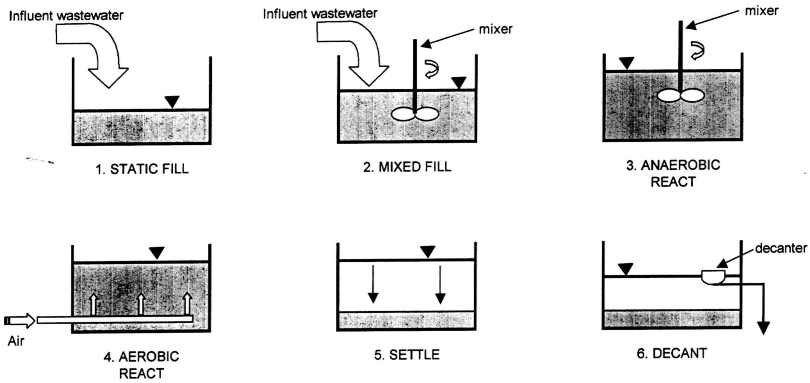

Figure 2.2: Suggested SBR Operation for Carbon, Nitrogen and Phosphorous Removal

Source: Arora, 1985

Biological phosphorus removal can be achieved in SBRs by creating a sequence of anaerobic conditions followed by aerobic conditions if readily biodegradable COD is present during the anaerobic phase (Goronszy, 1992). Biological phosphorus removal will occur in SBRs when the operating cycle begins with an anoxic period to eliminate nitrates followed by an anaerobic period to induce phosphorus release. Effluent phosphorus concentrations less than 0.5 mg/l have been reported in laboratory research and full-scale facilities have produced effluents with less than 1 mg/l total phosphorus without supplemental chemical addition (WEF, 1992). **Table 2.3** presents operational and performance data from some operating SBR plants in the United States.

Table 2.3: SBR operational and performance data

	Oak Point, Michigan	**Grundy Center, Iowa**	**Culver City, Indiana**	**Armada, Michigan**	**Manchester, Michigan**
Mean Cell Retention Time, days	51	-	15-45	39	15-60
MLSS/MLVSS, mg/l	3268/2402	1900/1140	2300/1380	3186/2235	2663/1820
F:M, l/day	0.04	0.10	0.12	0.06	0.034
Influent					
BOD, mg/l	126	200	100	157	74
T-P, mg/l	6.1	8.2	5.6	5.5	2.13
NH_4-N, mg/l	12.0	18.3	18.4	18.3	11.1
Effluent					
BOD, mg/l	5.0	5.0	9.2	10.3	3.0
T-P, mg/l	2.0	4.3	0.6[a]	0.48	0.50
NH_4-N, mg/l	0.4	0.5	1.0	3.43	0.43
NO_3-N, mg/l	3.7	3.5	1.3	-	-
Cycle time, hour	7.5	9	6	7	7.8
Number of units	2	2	2	3	3
Cycle breakdown: anoxic/aerobic/ settle/draw/idle, hour	2/3.3/ 0.8/0/3/1	2.8/3/ 1/1.3/0.8	3/0.5/ 0.7/0.7/1	2/1.5/ 1.0/0.5/2.0	4/1.5/0.75/ 0.75/0.8

[a] achieved with chemical addition.

Source: WEF, 1992

2.2 SBR Technology

The SBR technology is one of various methods to gain control over structure and functions of the microbial community in a multi-purpose bioreactor exposed to varying influent conditions. Typically, the reactor volume varies with time in the SBR, but remains constant in continuous flow systems. In contrast to continuous flow activated sludge systems, both biological reactions and biomass separation take place in the same tank. The SBR system is sometimes referred to as a single tank reactor (Irvine *et al.*, 1997).

The SBR is filled and drawn within a defined period of time. The Fill, React, Settle and Draw phases of the SBR are repeated continuously in a cycle. The cycle ends with Draw phase, or with an optional Idle phase. The continuous repetition of the cycle ensures that the micro-organisms in the wastewater are periodically exposed to a defined and regulated variation of process conditions (Irvine *et al.*, 1997).

The SBR provides benefits beyond the simple flexibility of varying the reaction period to improve contaminant removal. Irvine *et al.* (1997) found that, by controlling the cycle times, flow rates, nutrient availability and the availability of oxygen, the SBR has the ability to apply environmental pressures on a microbial consortium. Environmental pressures include feast conditions (high growth rate environment, aerobic) and famine conditions (low growth rate environment, anaerobic). Feast conditions would be the Fill and React phases, when organic substrate and oxygen is added to the MLSS in the tank, and aerobic micro-organisms will dominate. Likewise, famine conditions would prevail during the Settle and Decant phases, when anaerobic bacteria will thrive. During the start-up, the environmental pressures applied will enrich a given consortium. These bacteria will be those organisms which can utilize available carbon for energy and growth as well as survive in the environment created in the reactor. After enrichment, further changes in the operating environment will cause either changes in the physiological state of the organism selection and hence the bacterial population will change accordingly (Irvine *et al.*, 1997).

It was reported by Irvine *et al.* (1997) that a substantial increase in the initial substrate concentration is required to favor growth of floc forming organisms over filamentous organisms. Filamentous organisms are discouraged in systems that have high growth rates (i.e. at elevated substrate concentrations) and prolonged starvation periods. In general, organism selection depends upon a number of factors including the magnitude of the maximum growth rate achieved and the total time during which that rate was maintained, the magnitude and extent of starvation, and the frequency with which such feast and famine conditions occur (Irvine *et al.*, 1997).

2.2.1 Time Sequencing

Most of the SBR plants consist of two or more identically operated tanks that provide for the time sequencing of operations such as equalization, biological conversion, sedimentation and clarification during a complete reactor cycle. Some systems employ mixing and/or aeration to keep the micro-organisms and other solids in suspension while the reactor is filling. Mixing and/or aeration are turned off to allow for clarification during a quiescent settle period in suspended growth system (activated sludge). A clear supernatant is withdrawn from the reactor and an active culture remains in the reactor for the beginning of the next cycle (Irvine *et al.*, 1997).

The SBR is filled and drawn within a defined period of time. After completion of the fill phase, variations in the influent of the treatment plant no longer have any effect on the processes taking place in the reactor just filled except to limit or extend the total time allowed for the processes to take place. Once the duration of the Fill phase is specified, the time of the aerated and non-aerated React phase (i.e. the phase following Fill but before Settle) can be selected to achieve specific process goals. Similarly the time for the Settle and Draw phases can also be selected to achieve specific process goals (e.g. sludge settleability, size of the population of nitrifiers, denitrifies, Bio-P-bacteria, etc.) (Irvine *et al*, 1997).

The SBR achieves within a framework of time what a conventional system would achieve in terms of space. The application of microprocessor controls have made the time sequential operations relatively easy. The characteristics and performance of SBR systems have been extensively reviewed by Irvine and Ketchum (1989). Several full-scale SBRs are operating successfully in USA and UK for the treatment of sewage.

2.2.2 Sludge Wasting

The excess activated sludge produced each day must be wasted, or removed, to maintain a given food-to-microorganism ratio or mean cell-residence time. In a conventional activated sludge system, the common practice is to remove sludge from the return sludge line because it is more concentrated and requires smaller waste sludge pumps. An alternative method is to withdraw mixed liquor directly from the aeration tank. The waste sludge is then discharged to the sludge thickening facility. (Tchobanoglous and Burton, 1991).

Sludge wasting is not included as one of the five basic process steps because there is no set time period within the cycle dedicated to wasting. The amount and frequency of sludge wasting is determined by performance requirements, as with a conventional continuous-flow system. In an SBR operation, sludge wasting usually occurs during the settle or idle phases (Tchobanoglous and Burton, 1991).

A unique feature of the SBR system is that there is no need for a return activated-sludge (RAS) system. Because both aeration and settling occur in the same chamber, no sludge is lost in the react step and none has to be returned from the clarifier to maintain the sludge content in the aeration chamber (USEPA, 1987).

2.2.3 Nutrient Removal

A number of modifications have been made in the duration associated with each step to achieve specific treatment objectives (USEPA, 1987). Concern over excessive nutrient discharges to natural water systems and more stringent regulations has led to modifications in SBR systems to achieve nitrification, denitrification and biological phosphorus removal (USEPA, 1992).

Farrimond and Upton (1993) noted that, in the UK, the implementation of the Urban Waste Water Directive 1991 (UWWD) meant a tightening of discharge standards at many sites. At those sites classified as discharging to sensitive watercourses, limits for nutrient removal (N and P) were imposed and for the largest works of above 100,000 PE, implementation was required by 1998. The UWWD classified some receiving water courses and bodies of water as "sensitive" and limits for nitrogen and phosphorus were applied to these works discharging flows from contributing populations larger than 10,000. The standards were set as follows:

	Population Equivalent		% Reduction
	10,000-100,000	Over 100,000	
Total P (mg/l)	2	1	80
Total N (mg/l)	15	10	70-80

Source: Farrimond and Upton, 1993.

These limits are expressed as annual averages. Alternatively for Total Nitrogen, the daily average should not exceed 20 mg/l.

Biological phosphorus removal can be achieved in SBRs by creating a sequence of anaerobic conditions followed by aerobic conditions if readily biodegradable COD is present during the anaerobic phase. Biological phosphorus removal will occur in

SBRs when the operating cycle begins with an anoxic period to eliminate nitrates followed by an anaerobic period to induce phosphorus release. Effluent phosphorus concentrations less than 0.5 mg/l have been reported in laboratory research and full-scale facilities have produced effluents with less than 1 mg/l total phosphorus without supplemental chemical addition. The sequence of processes shown in **Figure 2.2** will accomplish carbon, nitrogen and phosphorus removal. In this configuration, phosphorus release and BOD uptake will occur in the anaerobic react phase with subsequent phosphorus uptake, carbon oxidation, and nitrification in the aerobic react phase (WEF, 1992).

Nitrification and Denitrification

Changes in nitrogen in a biological-treatment process are shown in **Figure 2.3**. Nitrification is the biological oxidation of ammonia to nitrate with nitrite formation as an intermediate. The micro-organisms involved are the autotrophic species *Nitrosomonas* and *Nitrobacter*, which carry out the reaction in two steps:

$$2NH_4^+ + 3O_2 \longrightarrow 2NO_2^- + 4H^+ + 2H_2O \quad (\textit{Nitrosomonas})$$

$$2NO_2^- + O_2 \longrightarrow 2NO_3^- \quad (\textit{Nitrobacter})$$

It is generally accepted that the specific growth rate of *Nitrobacter* is higher than the growth rate of *Nitrosonomas* and hence there is no accumulation of nitrite in the process and the growth rate of *Nitrosonomas* will control the overall reaction. (Eckenfelder, 1989a).

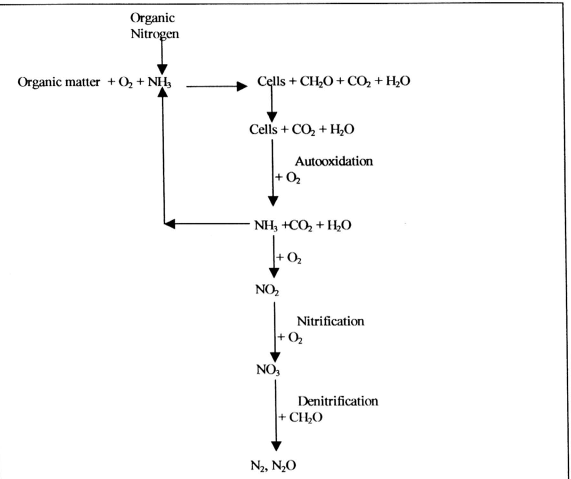

Figure 2.3 : The Nitrogen Cycle in Biological-Oxidation Processes

Source: Eckenfelder, 1989a.

Biological denitrification is achieved under anoxic (absence of molecular oxygen) conditions by heterotrophic micro-organisms that utilize nitrate as a hydrogen acceptor when an organic energy source is available. Denitrification will also occur under conditions of endogenous respiration, although at a much slower rate:

$$NO_3^- + \text{substrate} \longrightarrow N_2 + CO_2 + H_2O + OH^- + \text{cells}$$

Nitrate nitrogen is converted to reduced forms such as N_2, N_2O and NO. The breakdown of carbonaceous organics in the denitrification reactions is similar to that in the aerobic process. Anoxic conditions are needed in a denitrifying system. It is necessary to provide a sufficiently long solids retention time to ensure the growth of nitrifying organisms: a sufficient active fraction of biomass at a dissolved oxygen level adequate for nitrification to occur and an anaerobic or anoxic phase of sufficient duration to permit nitrate reduction (Eckenfelder, 1989a).

Nitrification is the initial step in the removal of nitrogenous compounds from wastewaters. It involves the two-step conversion of ammonia to nitrite (ammonia oxidation) and nitrite to nitrate (nitrite oxidation) (Halling-Sørensen, 1993). Denitrification of the nitrate to nitrogenous gas removes the nitrogen from solution (Robertson and Kuenen, 1991). If nitrogen removal fails, the nitrogenous compounds passing into waterways may cause a series of environmental and medical problems (Argaman, 1991). Although autotrophs comprise only a small percentage of the mixed liquor community in wastewater treatment systems, they are responsible for the bulk of nitrification (Robertson and Kuenen, 1991; Randall, 1992a; Randall, 1992b). In wastewater treatment systems, *Nitrosomonas* (an ammonia oxidizer) and *Nitrobacter* or *Nitrospira* phylum (nitrite oxidizers) are the autotrophs presumed to be responsible for nitrification (Halling-Sørensen and Jørgensen, 1993).

Availability of substrate or carbon source is an important element in BNR process. If there is sufficient substrate in aeration tank, nitrification may not be fully developed since most of oxygen would be taken by the more competitive heterotrophs (Oleszkeiewicz and Berquist, 1988). When denitrification following nitrification occurs, availability of carbon source is an important factor to reduce the oxidized form of nitrogen to nitrogen gas. A release of phosphorus in anaerobic conditions must be followed by phosphorus uptake in aerobic condition for BNR.

The degrees of nitrification and denitrification are affected not only by the availability of organic substrate but also by operating temperature. It is generally known that the nitrification rate significantly decreases at low temperature (Randall, 1992b). It is known that the activity of denitrifiers is seriously hampered by low temperature (Oleszkeiewicz and Berquist, 1988). For instance, Henze (1991) reported that the denitrification rate at 8°C is about 7 times less than that at 30°C. On the other hand, temperature effect on biological phosphorus removal is not fully understood.

The concentration of oxidized nitrogen affects the phosphorus release in anaerobic zone. It has been reported that the concentration of oxidized nitrogen in the recycled flow must be less than 10 mg/l (Tetrualt *et al.*, 1986) and less than 3 mg/l for more diluted wastewater at 20°C (Lee, 1995). If phosphorus is not released because of the nitrate recycle, an aerobic zone would increase. Excess amount of nitrate recycle could hamper the biological phosphorus removal mechanism.

In their review on nitrogen removal as an important aspect of the wastewater treatment process, Van Loosdrecht and Jetten (1998) noted that the large amount of nitrogen compounds involved, the numerous reactions that can occur (**Figure 2.4**) and the slow growth rate of many of the bacteria involved hampers the study of nitrogen conversion processes in activated sludge.

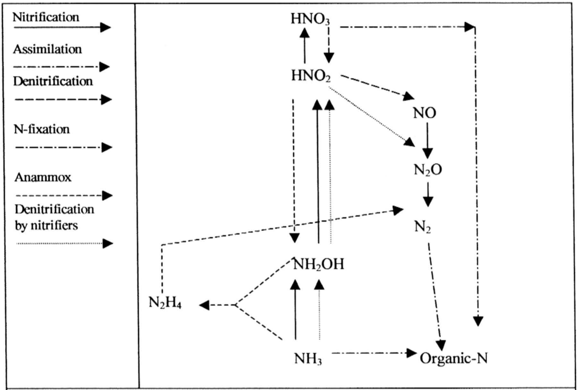

Figure 2.4: Possible Microbial Conversions of Nitrogen

Source: Van Loosdrecht and Jetten, 1998

Autotrophic Nitrification

Nitrification is generally performed by autotrophic or mixotrophic (Laanbroek *et al.*, 1994) micro-organisms. In two steps ammonium is oxidized via nitrite to nitrate. There have been a number of different types of nitrifying bacteria identified. These either oxidize ammonium to nitrite or nitrite to nitrate. There is no organism described which can directly oxidize ammonium to nitrate. The oxidation of ammonium is generally attributed to *Nitrosomonas europaea*, and the oxidation of nitrite to *Nitrobacter agilis*.

For N-removal processes it is beneficial if ammonium is only oxidized to nitrite and thereafter denitrified. This has been the subject of many investigations. Most often mentioned is the possibility to inhibit the nitrite oxidizers by a change in pH (e.g. Turk and Mavinic, 1989; Abeling and Seyfrie, 1992). Since ammonia and nitric acid are the toxic compounds (and substrates) a small change in pH will have a rather strong effect on the concentration of these compounds (Hellinga *et al.*, 1997). A changed pH usually results initially in nitrification towards nitrite, however after some time (sometimes weeks) adaptation occurs and full nitrification is restored (Alleman and Irvine, 1984). In general the inhibition of nitrite oxidizers therefore does not seem to be stable for long-term operation.

A second opinion is the wash-out of nitrite oxidizers based on growth rate. At elevated temperature (>15 °C) the ammonium oxidizing bacteria have a higher growth rate than the nitrite oxidizers. Carefully controlling the sludge age has been shown to be a good operating parameters for a stable partial nitrification. In practise, this procedure can be used for nitrification at temperatures above 20°C (Hellinga *et al.*, 1997).

Finally substrate competition can be used to out-compete nitrite oxidizers. The lower affinity of oxygen for nitrite oxidizers compared to ammonium oxidizers can be used to selectively restrict the growth of nitrite oxidizers (Turk and Mavinic, 1989; Hanaki *et al.,* 1990; Laanbroek and Gerards, 1993). Denitrification of nitrite will further decrease the growth of nitrite oxidizers. This can be achieved by creating anoxic conditions inside the flocs or by a rapid circulation between aerobic and anoxic conditions. Operating the treatment plant at a low dissolved oxygen concentration can therefore lead to a stable situation where nitrite oxidizers are out-competed. However, these conditions can lead to decline of the sludge volume index.

Denitrification

Denitrification is the reduction of nitrate towards nitrogen gas. This conversion has many intermediates such as HNO_2, NO, and N_2O. Denitrification requires an electron donor, which can be organic material or reduced compounds such as sulphide or hydrogen. Of concern is the release of above-mentioned intermediates from the treatment process into the environment (Von Schultess *et al.,* 1994; Czepiel *et al.*, 1995). Under electron donor elimination these intermediates can be readily formed. At low dissolved oxygen concentrations the different enzymes in the denitrification metabolism might be inhibited differently. Moreover, when cells are subject to transitions between aerobic and anoxic conditions, formation of these intermediates is enhanced (Otte *et al.*, 1996; Van Bentum *et al.*, 1998). The latter might be due to the enzyme regulation inside the cell (more than 40 genes are involved in the denitrification process) which cannot react immediately to a changed environment (Baumann *et al.*, 1996, 1997).

An important aspect concerning the possible emission of N_2O into the environment during denitrification is its high solubility in water (Eckenfelder, 1989a). This means that N_2O is not easily stripped into air and that an initial accumulation can be followed by consumption when the bacteria have been adapted from the change from aerobic to anoxic conditions (in general several minutes).

Denitrification by autotrophic nitrifiers

Autotrophic nitrifying bacteria (such as *Nitrosomonas europaea*) can produce significant amounts of N_2O, NO or N_2 (Hellinga *et al.*, 1997). However the reported conversion rates of $5x10^{-4}$ (Anderson and Levine, 1986), or $1.4x10^{-5}$ gN/gVSS.d (Bock *et al.*, 1995) are an order of magnitude slower than conversion rates for conventional nitrification and denitrification reactions (approx. 1-5 gN/gVSS.h).

Anaerobic ammonium oxidation (Anammox)

Conventional ammonium oxidation occurs by the action of an enzyme, ammonium mono-oxygenase, which requires molecular oxygen. Ammonium removal in an anaerobic fluidised denitrifying bed, treating effluent from a methonogenic reactor was observed (Mulder *et al.*, 1995). This is known as anaerobic ammonium oxidation (Anammox). The autotrophic organisms catalyse two peculiar conversions: anaerobic oxidation of ammonium to nitrogen gas, and anaerobic oxidation of nitrite to nitrate. Due to its slow growth rate, 0.1 to 0.05 day^{-1}, it is highly unlikely that these organisms will appear in normal heterotrophic denitrifying wastewater processes. (Van de Graaf, *et al.*, 1996)

Limiting factors

The limiting factors with respect to nitrogen removal were temperature, rate of nitrification and the availability of sufficient carbon and energy sources for denitrification. (Harremoes *et al.*, 1998)

Henze *et al.* (1995) noted that nitrification is based on slowly growing bacteria and that the main condition of nitrogen removal based on nitrification-denitrification is an aerobic sludge age of the activated sludge system in excess of the limiting value, below which the nitrifiers are washed out of the system at a faster rate than they can grow. At the same time, the rate of growth is temperature dependent.

Pilot plant studies showed that the nitrification was partly inhibited (Hansen and Nielsen, 1992; Grüttner *et al.*, 1994) and that the major contributor of inhibitory substances was the industry. A high degree of inhibition was detected in areas with industrial activity, whereas no or minor inhibition was observed in residential and office areas. However, it is found that a large variety of different substances contributes to the inhibition.

In tests with inoculation of adapted sludge, the maximum nitrification capacity was obtained 2 months after start-up, whereas in the tests without inoculation, it took 3 to 7 months (Sinkjær *et al.*, 1996). This indicates that the nitrifiers can adapt to a certain extent to the amount of inhibitory substances in the influent.

Biological Phosphorus Removal

Phosphorus, like N, is assimilated by bacteria and is removed from the water with the excess sludge. The degree of removal is related to the rate of sludge production. The basic principal of bio-P removal is shown in **Figure 2.5,** whereby the bacteria are exposed alternatingly to anaerobic and aerobic conditions (Arvin, 1985).

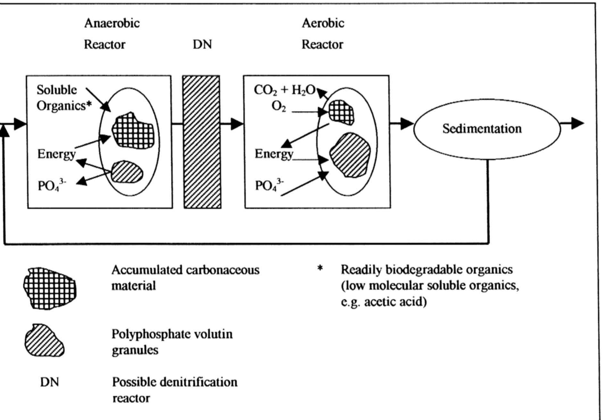

Figure 2.5: Biological Release and Uptake of Phosphate while Alternating Between Anaerobic and Aerobic Conditions

Source: Arvin, 1985

Eckenfelder (1989b) reported that certain bacteria, notably *Acinetobacter*, posses the ability to absorb low-molecular-weight organics (e.g. fatty acids) under anaerobic conditions. The energy required for this is made available by the release of phosphorus bound as polyphosphates in volutin granules in the protoplasm of the bacteria. Under subsequent exposure to aerobic conditions, the organic matter is oxidized and energy is made available for growth and for the re-accumulation of phosphates into polyphosphates in the bacteria. The net effect is an excess content of phosphorus in the bacteria possessing this ability. Due to their ability to hoard the readily available organic matter for their own consumption, these bacteria have a competitive edge over other bacteria. Under proper process conditions, they can flourish and dominate the population, resulting in an increased P content in the waste sludge (Eckenfelder, 1989b).

Rensink (1991) found that the circulation of activated sludge through anaerobic and aerobic zones is the basis for biological phosphorus removal processes. Anoxic zones are sometimes incorporated for denitrification, mainly because the presence of nitrate is assumed to inhibit phosphorus removal (Rensink, 1991).

Kuba *et al.*, (1993) used an anaerobic-anoxic SBR (A_2 SBR) in order to investigate the possibility of phosphorus removal utilizing nitrate as an electron acceptor instead of oxygen in biological phosphorus removal processes. The reactor was supplied with synthetic wastewater, in which acetic acid (HAc) and phosphate were added at concentrations of 400 mg-COD/l and 15 mg-P/l. A conventional anaerobic-aerobic

SBR (A/O SBR) was also operated to compare with the anaerobic-anoxic SBR. **Figure 2.6** is a schematic diagram showing the two SBR operations used (Kuba *et al.*, 1993).

The anaerobic-anoxic SBR operation resulted in a stable phosphorus removal and accumulation of phosphorus removing bacteria using nitrate as an electron acceptor (Kuba *et al.*, 1993). Immediately after inoculation from a phosphorus removing plant (Renpho system) phosphorus uptake was observed, indicating that phosphorus removing bacteria which are able to utilize nitrate were already present in conventional phosphorus removing sludge. All HAc was consumed during the anaerobic phase, and around 100 mg-P/l phosphorus was released in both SBRs operation. The released phosphorus was completely removed during the aerobic or anoxic, but phosphorus leakage occurred at the end of the anoxic phase in the A_2 SBR. The phosphorus leakage resulted in a lower degree of overall phosphorus removal efficiency in the A_2 SBR . It was found that leakage did not occur if NO_3^- was present. Kuba *et al.* (1993) concluded, therefore, that nitrate addition should be controlled carefully. This could be done by using the redox potential signal.

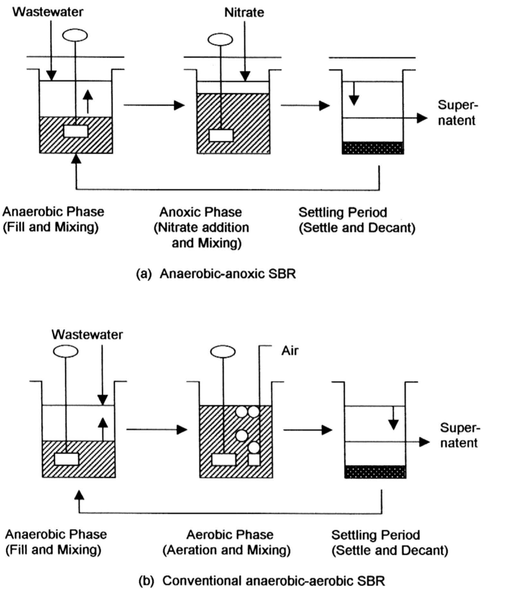

Figure 2.6: Schematic Diagram of SBR Operation (Anaerobic-Anoxic and Anaerobic-Aerobic)

Source: Kuba *et al.*, 1993

In order to find temperature effects in biological phosphorus removal, Baetens, *et al.* (1999) operated an SBR in an anaerobic-aerobic sequence to cultivate an enriched biological phosphorus removing sludge. The impact of long-term temperature changes on the stoichiometry and kinetics of the different processes involved was studied at 20, 15, 10 and 5°C. At 20, 15 and 10°C complete phosphorus removal was achieved over long periods. Maximum anaerobic phosphorus concentrations of 140 mg/l were recorded. For this temperature range, acetate was always fully consumed, whereas at 5°C, acetate broke through to the aerobic phase. Nitrification was observed at 20, 15 and 10°C, whereas at 5°C nitrifiers were washed out. Nitrate was never observed anaerobically. Baetens, *et al.* (1999) inferred that during the 3 minutes filling period, all nitrate present from the preceding aerobic period was already consumed.

Studies carried out by Cuevas-Rodriguez, *et al.*, 1998, using an activated sludge SBR fed with pre-fermented wastewater showed that :

- Volatile Fatty Acid (VFA) concentrations of up to 223 ± 24 mg/l can be achieved with an anaerobic SBR without pH neutralization. The VFA species were acetic acid (63%), propionic acid (25%), and butyric acid (12%).
- Fermentation reduces the particulate COD increasing the dissolved COD.
- The presence of VFA in the wastewater increases the capacity of bacteria to accumulate phosphates. The three tested organic loading rates of 0.13, 0.25 and 0.35 $kgCOD_{total}$/kg SS per day produced effluent PO4-P concentrations under 1.1 mg/l.

- The effluent COD remained between 37 and 38 mg/l, independently of the organic load.
- Nitrification was present during the whole experimentation and was not affected by the presence of VFA in the influent.
- Denitrification was observed during the filling and anoxic/anaerobic phases. Inorganic nitrogen removal was 88, 66 and 81 percent for the 0.13, 0.25 and 0.35 $kgCOD_{total}/kg$ SS per day organic loading respectively.
- The settleability of the sludge increased with the organic load. Sludge Volume Index (SVI) is inversely proportional to the organic load. The lower the SVI, the higher the settleability. Sludge volume index values in ml/g were 71, 50 and 45 ml/g for the 0.13, 0.25 and 0.35 $kgCOD_{total}/kg$ SS per day organic loading respectively. All values of settleability were under 80 ml/g, which is considered excellent.

It was found that phosphate concentrations in the effluent are lower when the removal is associated with VFA in the influent. Phosphate concentration is high in the anoxic-anaerobic phase, but reduces significantly in the aerobic phase. Nitrates are present in the influent. Denitrification (reduction of nitrates) takes place during the anoxic-anaerobic phase. Denitrification processes begin during the fill phase and complete denitrification is achieved during the first hour of the anaerobic phase. Nitrates are removed within the first hour after filling the reactor and the production begins immediately after the establishment of the aerobic conditions. Ammonium increases slightly during anoxic-anaerobic phase, then slowly decreases through the aerobic stage. As a product of the nitrification processes, ammonium is consistently consumed during the aerobic phase at the same time that nitrates are produced. Higher organic loading rates provides higher VFA amounts and higher capacity of the bacteria to accumulate phosphates. The amount of polyphosphates can contribute to higher relative densities and faster settling speed of the sludge. These findings are summarised diagrammatically in **Figure 2.7** (Cuevas-Rodriguez, *et al.*, 1998).

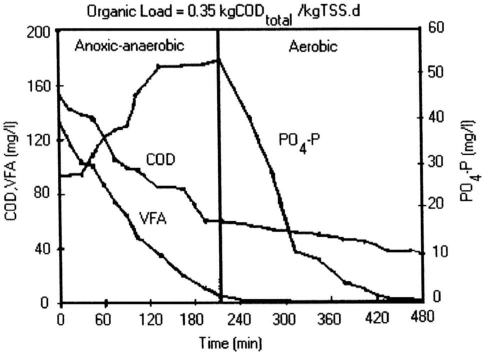

Figure 2.7: VFA, Dissolved COD and PO_4-P Profiles for Treatment of Wastewater Using SBR

Source: Cuevas-Rodriguez, *et al.*, 1998

Various researches (e.g. Chiesa and Irvine, 1985; Wanner, 1992; Casey *et al.*, 1994) demonstrated experimentally that frequent shifting of activated sludges between feast and famine conditions is a very effective means to control excess growth of filamentous organisms. Frequent shifting of activated sludges between aerobic, anoxic and anaerobic zones allows establishment of microbial communities capable of executing nitrification, denitrification and enhanced biological phosphate uptake.

Ubukata and Takii (1994) demonstrated that the enzyme system responsible for EBPR was inducible after at least two anaerobic/aerobic cycles. In order to induce the necessary enzymes for EBPR, the cells need to be incubated under aerobic conditions without any organic substrate. EBPR organisms have a very long generation time i.e. low specific growth rate (Nakamura *et al.*, 1991; Okada *et al.*, 1992; Ubukata and Takii, 1994).

Nakamura *et al.* (1991) demonstrated the ability for high poly-P accumulation of a Gram-positive bacteria when the culture was maintained under "micro-aerobic" conditions with a high concentration of organic substrate for 15 days. They postulated that the enzyme system of EBPR might be induced during the "micro-aerobic" condition. Hence this would indicate the importance of aerobic conditions on enzyme induction.

Despite denitrification and nitrate ammonification being thought to be anoxic processes, this classic theory was found to be not absolute. An increasing number of bacteria are known to denitrify in the presence of oxygen. Carter *et al.* (1995) demonstrated that the periplasmic nitrate-reductase provided the necessary biochemical apparatus for aerobic nitrate respiration. It was suggested that the co-respiration of nitrate and oxygen might offer a physiological advantage in environments subjected to a fluctuating oxygen availability (Patureau *et al.*, 1997).

Tonkovic (1998) observed that bacteria often alternate between phases of growth and non-growth because of fluctuations in substrate availability in their local environment. When growth substrates become limiting, bacteria generally initiate starvation responses. These phenotypic changes are integrated parts of survival strategies in which survival time is often optimised by lowering energy metabolism while maintaining some basic cellular processes. Feast-famine regimes enrich for organisms which use storage polymers to balance their growth. Polymer formation leads to a high sludge yield provided the polymer is not oxidized. Storage polymer formation and subsequent growth on it leads to a lower sludge yield compared to direct growth on the soluble COD (Tonkovic, 1998).

To remove nitrogen and phosphorus simultaneously from wastewater, Hamamoto *et al.* (1997) carried out studies on the Intermittent Cyclic Process (ICP), also known as the SBR. It alternates aerobic and anaerobic conditions periodically in a single reactor. These studies showed that high levels of nutrient removal (nitrogen and phosphorus) were possible by decreasing the aeration time ratio (aeration time to total cycle time). The aeration time ratio can be defined as follows:

$$\text{Aeration time ratio} = \frac{\text{Total aeration time in one cycle (hours/cycle)}}{\text{Length of one cycle (hours/cycle)}}$$

The ICP is a batch activated sludge treatment process, which consists of four steps: (i) mixing, (ii) aeration, (iii) settling and (iv) drawing. An ICP treatment plant uses two tanks which operate at opposite phases of the cycle. In a typical cycle, 45 minutes of mixing is followed by 15 minutes of aeration. This is repeated three times. Influent flows into the tank throughout the 3 hours. This is followed by 1 hour of settling and 2 hours of drawing. While the settling and drawing occur in one tank, influent fill, mixing and aeration take place in the second tank. The two tanks are always at opposite stages of the cycle, which allows for the uninterrupted flow of wastewater into one of the two tanks. The experiment conditions and shown in **Table 2.4** and the operation conditions are shown in **Figure 2.8.** Each tank completes 4 cycles per day.

The operating conditions and corresponding aeration time ratios were as follows: 6 hours of continuous aeration 1 (Run 1-1); 3 hours of aeration, 0.5 (Run 1-2); 15 minutes of mixing and 45 minutes of aeration, repeated 3 times, 0.375 (Run 1-3); 30 minutes of mixing and 30 minutes of aeration, repeated 3 times, 0.25 (Run 1-4 and Run 3-1); 40 minutes of mixing and 20 minutes of aeration, repeated 3 times, 0.167 (Run 3-2); 45 minutes of mixing and 15 minutes of aeration, repeated 3 times, 0.125 (Run 1-5 and Run 2-1); and automatic fuzzy logic control of mixing and aeration time (Run 3-3). For all runs settling was 1 hour and drawing was 2 hours.

Table 2.4: Experiment Conditions for Intermittent Cyclic Process

		Laboratory Studies					Pilot Study	Full scale studies		
Run no.		1-4	1-2	1-2	1-4	1-5	2-1	3-1	3-2	3-3
Water temperature	(°C)	20	20	20	20	20	11-24	16-24	14-25	22-27
No. of cycles	(cycles per day	-	4	4	4	4	4	4	4	4
Flow rate	(L or m^3 per day)	20L	20L	20L	20L	20L	$18m^3$	339-394 m^3	341-474 m^3	427- 477 m^3
Volume of reactor	(L or m^3 at HWL)	20L	20L	20L	20L	20L	$18m^3$	650 m^3	650 m^3	650 m^3
	(L or m^3 at LWL)	-	15L	15L	15L	15L	$12m^3$	494 m^3	494 m^3	494 m^3
Aeration time	(minutes per cycle)	360	180	45x3	30x3	15x3	15x3	30x3	20x3	Varia-ble
Mixing time	(minutes per cycle)	-	-	15x3	30x3	45x3	45x3	30x3	40x3	Varia-ble
Aeration time ratio	-	1	0.5	0.375	0.25	0.125	0.125	0.25	0.167	Varia-ble
BOD-SS load	(gBOD/gSS per day)	0.096	0.090	0.085	0.083	0.083	0.059	-	-	-

Source: Hamamoto *et al.*, 1997

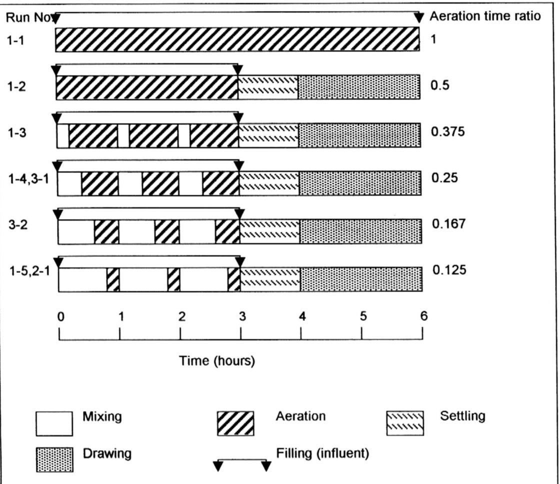

Figure 2.8: Operating Conditions for Intermittent Cyclic Process for Simultaneous Nitrogen and Phosphorus Removal

Source: Hamamoto *et al.*, 1997

The composition of the synthetic wastewater which was used in the laboratory studies is shown in **Table 2.5.**

Table 2.5: Composition of Synthetic Wastewater (Laboratory Studies)

Composition	Concentration (mg/l)
Peptone	180
Meat Extract	120
$Na_2HPO_4 \cdot 12H_2O$	25.2
$CaCl_2 \cdot 2H_2O$	5.56
$MgSO_4 \cdot 7H_2O$	6.2
KCl	4.2
NaCl	90
$NaHCO_3$	250
BOD*	169
T-N*	35.8
T-P*	4.5

* measured value

Source: Hamamoto, *et al.*, 1997

In the said study, (Hamamoto *et al*, 1997) the Total-N removal rate decreased sharply as the aeration time ratio increased, and fell to about 10% when the aeration time ratio was increased to 1 in the continuous aeration process. Similarly the phosphorus removal rate decreased very rapidly as the aeration time ratio increased. **Figure 2.9** shows the relationship between the aeration time ratio and the removal rate of nitrogen and phosphorus. No phosphorus removal occurred when the aeration time ratio was 1. These results show that when the aeration time ratio increased NO_x-N (= NO_2-N+ NO_3-N) was not sufficiently denitrified in the mixing stage. Therefore the remaining NO_x-N prevented the release of phosphorus and the Total-P removal rate fell (Hamamoto *et al*, 1997).

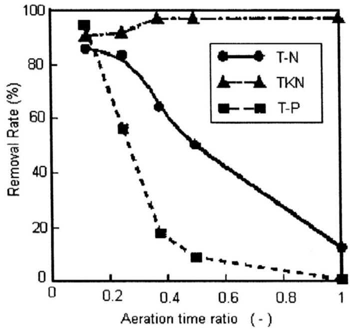

Figure 2.9: Relationship between aeration time ratio and nitrogen and phosphorus removal rate (Laboratory studies, Runs 1-1 to 1-5)

Source: Hamamoto *et al.*, 1997

In the case of the pilot plant operation, the removal of organic substances was good. Influent BOD values were in the range of 170-200 mg/l. Effluent BOD values were below 20 mg/l, and the average was 8.1 mg/l. Influent T-N values were between 30-42 mg/l. Effluent T-N values were below 10 mg/l, and the average was 5.0 mg/l. The mean nitrogen removal rate was 86%. Influent T-P values ranged between 4-5 mg/l. Effluent T-P values were mostly below 1.5 mg/l, and the average was 0.79 mg/l. The mean phosphorus removal rate was a relatively high 82%.

In the studies carried out by Hamamoto *et al.* (1997), performance of the SBR was further improved with the recently developed controller which uses "fuzzy logic" to automatically determine optimal mixing and aeration periods. Mixing and aeration time were automatically set by the fuzzy controller based on readings of DO, pH, ORP and water level in the batch reactor. Average nitrogen and phosphorus removal rates in the full-scale plant were 96% and 93%. The results showed that nitrogen and phosphorus removal improved when the aeration time ratio was low and that the fuzzy control was more effective than timed control.

Muñoz-Colunga and González-Martínez (1996) operated a pilot biofilm SBR for 400 days in order to determine the effects of different operation strategies on the capacity of the biofilm to remove nutrients (C, N and P). The reactor was fed with wastewater from the main campus of the National University of Mexico. The raw wastewater characteristics are shown in **Table 2.6.** The wastewater was enriched with a molasses and phosphate solution when needed to vary the nutrients concentration. The operation was controlled with an industrial programmable timer. The treatment cycles were adjusted with four stages: filling, anaerobic phase, aerobic phase and draw of treated wastewater. Cycles of 8 hours and 12 hours were tested with different anaerobic/aerobic time rates. The highest removal COD and PO_4-P rates were obtained with 12-hour cycles and phases duration of 37/63 percent anaerobic/aerobic. Analyzing COD, TOC, PO_4-P, NH_4-N and NO_3-N, shifting of the different bacterial groups could be followed. When the organic loading rate was higher than 5 gCOD/m^3.d the activity of the phosphate accumulating bacteria and nitrification could not be observed. The best results regarding phosphate removal and nitrification were obtained when the mean organic load was 3 $gCOD/m^3.d$.

Table 2.6: Raw wastewater characteristics

Parameter	Highest	Lowest	Average
Total COD (mg/l)	345	93	190
Dissolved COD (mg/l)	317	30	66
Total organic carbon TOC (mg/l)	77	9.6	17
PO_4-P (mg/l)	4.2	1.2	1.0
NH_4-N (mg/l)	13.6	2.2	7.0
NO_3-N (mg/l)	4.12	1.33	2.8
PH	8.0	7.1	7.6
Dissolved oxygen DO (mg/l)	4.5	1.0	3.2
Temperature (°C)	20.5	16	18.7

Source: Muñoz-Colunga and González-Martínez (1996)

The results of the study: a) COD and TOC removals increase with the cycle duration (the highest removal was achieved with 12-hour cycle and 63 percent aerobic duration): b) TOC and tCOD had similar removal and sCOD showed higher removal (10 percent more); c) the highest PO_4-P removals were achieved with 12-hour cycle and anaerobic phase of 50 and 35 percent: this means that longer aerobic periods improve PO_4-P removal; d) the highest PO_4-P released during the anaerobic phase was for the longest anaerobic phase: removal was not good because the aerobic time for the uptake was not long enough; e) organic load values under 4 $gCOD_{total}/m^3.d$ are a condition to achieve PO_4-P removal.

The co-existence of different micro-organisms presents a competitive relationship for oxygen between the phosphate accumulating bacteria (poly-P organisms) and the nitrifying bacteria. Hang-sik *et al.* (1993) and Rusten and Eliassen (1993) reported that in a SBR with suspended biomass, only partial nitrification was obtained when an excellent phosphate removal occurred and vice versa. However, González-Martínez and Wilderer (1991) reported efficient phosphate removal and compete nitrification using a biofilm SBR.

Garzon-Zuniga and González-Martínez, (1996) reviewed reported experiences for commercial industrial processes which remove nitrogen and phosphorus in activated sludge systems, and observed that the following behavior is expected in an SBR: a) after filling the reactor, an anaerobic phase leads to release of previously stored phosphates and storage of organic pollutants from the wastewater; b) an aerobic phase means that the phosphates in the liquid are captured and stored as polyphosphates (poly-P), nitrification takes place; c) an anoxic period generates denitrification to reduce nitrates and, eventually makes some phosphate to be released; d) finally, an aerobic, or oxic, phase makes the microorganism capture the rest of phosphates from the liquid and metabolize endogenous and exogenous organic material.

In the experiment by Shin and Jun (1992), excess phosphorus removal was achieved in a week on the proposed operating method which is to keep substrate zero before the start of aerobic react. The removal rates of both TOC and phosphorus were above

98% in one week of operation. Phosphorus removal bacteria can utilize glucose as well as acetate and also use nitrate as a single electron acceptor. The developed excess phosphorus removal system had good stability against the impact load.

The seed sludge was taken from a municipal wastewater treatment plant which did not have the excess phosphorus removal characteristics. In this experiment with a SRT of 8 days, phosphorus release and uptake appeared from the initial operations, but phosphorus removal mechanism was not improved for several weeks. The SBR operation cycle used was: 0.5 hour static Fill, 2.5 hour anaerobic React, 5 hour aerobic React, 2-hour Settle, 0.5 hour Draw and 1 hour Idle.

Figure 2.10 shows the profiles of pollutants in a cycle after 1 week operation. The removal characteristics of pollutants (TOC, phosphorus and ammoniacal nitrogen) shown in **Figure 2.10** indicate the methods to culture EPRB effectively. That is, substrate left after anaerobic React period helped the heterotrophs survive in the following aerobic React period.

To prevent heterotrophs from growing in the system, the remaining substrates were removed before starting aerobic React. The supernatant after 30 minutes settling was drawn out and replaced with the same feed except carbon sources. The removal of pollutants after changing the operation mode was enhanced significantly as shown in **Figure 2.11.** The released phosphorus in anaerobic React reached 50 mg/L as P which was 5 times higher than that of influent and was uptaken completely within 3-

hour aerobic React. As the substrate uptake rate during the anaerobic React was enhanced, the TOC after anaerobic React was decreased to 20 mg/l from 70-80 mg/l.

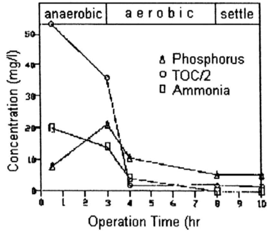

Figure 2.10: TOC, PO_4^{3-} -P, and NH_4^+-N after one week operation

Source: Shin and Jun, 1992

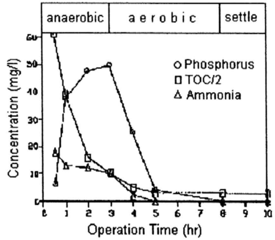

Figure 2.11: Performance of SBR after EPRS Development

Source: Shin and Jun, 1992

2.2.4 Settling Properties

Studies carried out by Wanner, 1992, comparing a continuous-flow cascade of 10 completely mixed tanks (System 1) with an anaerobic-aerobic activated sludge SBR system (System 2) showed that the continuous and SBR systems were comparable as far as COD, phosphorus and nitrogen removal are concerned, but the activated sludge from the SBR model exhibited slightly higher specific rates of nitrification and denitrification. The continuous and SBR differed significantly in the biocenoses of activated sludges developed in the system. While the continuous system produced activated sludge with medium-sized, irregular flocs with acceptable settling properties, the activated sludge from the SBR exhibited high settling velocities caused by giant flocs; zone settling velocities (ZSV) ranged between 4 and 7 m/h. The presence of these giant flocs eliminated the impact of filamentous micro-organisms, frequently observed in the biocenosis of activated sludge from SBR, on settling properties. The SVI values of the activated sludge from the SBR (System 2) were more stable, mostly below 100 mg/l (See **Figure 2.12**).

Wanner (1992) reported that the microscopic examination of the activated sludge during the cycles of the SBR showed that the intracellular granules in the observed filamentous micro-organism were synthesized in the anaerobic period and disappeared under aerobic conditions. The ability to accumulate and utilize storage products may explain the occurrence of the filamentous micro-organism in the SBR system. Contrast between excellent settling properties and frequent occurrence of filamentous micro-organisms in the biocenosis of activated sludge from SBR was probably caused by the existence of large and very compact flocs. The heavy flocs with a diameter of nearly 1 mm swept down the aggregates of filaments and smaller flocs. The formation of such giant flocs can be explained by the operational regime of the SBR. The flocculation was supported by gently stirring the mixed liquor during both anaerobic and aerobic period and by the quiescent conditions in the sedimentation period (Wanner, 1992).

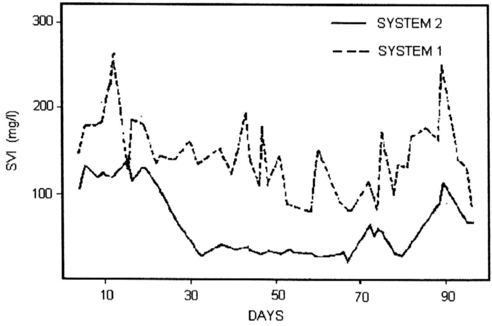

Figure 2.12: The Course of SVI – Comparison of Continuous Activated Sludge System and SBR

Source: Wanner, 1992

Filamentous bulking and the long sludge age required for nitrification are 2 important factors that limit the wastewater treatment capacity of biological nutrient removal activated sludge systems (Ekama and Wentzel, 1999). Observations from full-scale plants indicate support for the hypothesis that a significant stimulus for filamentous bulking in BNR systems is alternating anoxic-aerobic conditions with the presence of oxidized nitrogen at the transition from anoxic to aerobic.

From their experiments, Casey *et al.* (1994) concluded that the major factor influencing low F/M filament bulking was alternative anoxic-aerobic conditions forcing the heterotrophic organisms to switch between DO and nitrate as terminal electron acceptors. They formulated a hypothesis for the cause of low F/M filament bulking: if denitrification is not complete under anoxic conditions (i.e. all nitrate and nitrite not denitrified) the floc-formers, which denitrify nitrate fully to nitrogen gas via nitrite, nitric oxide and nitrous oxide are inhibited in the oxygen uptake under subsequent aerobic conditions by the denitrification intermediates accumulated under the preceding anoxic conditions, in particular nitric oxide (NO). This inhibition of the floc formers under aerobic conditions provides an advantage to the filaments which reduce nitrate only to nitrite and therefore are not inhibited in the oxygen uptake by nitric oxide (Ekama and Wentzel, 1999).

2.3 CYCLE OPERATION

Currently used SBR systems, designed for BOD and SS removal, have five basic steps in common that are carried out in the sequence as follows:

i) Fill,

ii) React (aerobic/anoxic),

iii) Settle (sedimentation/clarification),

iv) Draw (decant), and

v) Idle.

The designer or future operator can vary the time dedicated to each phase. Several different functions can occur during any one phase, depending upon the requirements of a particular treatment objective.

In SBR facilities requiring biological nutrient removal (Norcross, 1992), the traditional wisdom has been to use an "Anoxic Mixed Fill" at the beginning of the cycle to mix the raw influent with the biomass in an attempt to achieve the anaerobic conditions required for biological nutrient removal. It must be understood that at the end of the "React" period, all of the wastewater in the reactor has been treated; if a reactor is 4.9 meters deep, at the end of the "Settle" period, there will be a 1.2 meter sludge blanket, with 3.7 meters of treated effluent on top of it. After the "Decant" period, there will be a 1.2 meter sludge blanket, with 2.4 meters of treated effluent above. In an "Anoxic Mixed Fill" period, the raw influent is so diluted by the 2.4 meters of treated effluent that anaerobic conditions are seldom achieved. The influent distribution manifold eliminates the need to operate the mixing pumps during "Anoxic Fill"; raw influent is distributed evenly throughout the settled sludge without additional mixing. Anaerobic conditions are assured in the lower portion of the reactor, making nutrient removal much more consistent than in a conventional SBR.

2.3.1 Fill Phase

During the Fill Phase, raw or settled wastewater is fed into the reactor. Thus, Fill provides for the addition of influent and may be static, mixed or aerated. Static fill results in minimum energy input and high substrate concentration at the end of Fill. Mixed Fill results in denitrification, if nitrates are present. This denitrification results in a subsequent reduction of oxygen demand and energy input, giving rise to the anoxic or anaerobic conditions required for biological phosphorus removal. Aerated fill results in the beginning of aerobic reactions, a reduction of cycle time and holds substrate concentrations low. **Figure 2.13** is a schematic diagram of the various types of Fill (Ketchum, 1997).

If Mixed Fill is selected, the substrate concentration, dissolved oxygen concentration, and nitrate concentration vary during the Fill period. Assuming oxygen and nitrate are present in the SBR at the beginning of Mixed Fill, aerobic biological reaction would occur during the initial period of Mixed Fill, resulting in a reduction of dissolved oxygen and substrate. When oxygen is not available, nitrate will serve as the electron acceptor and anoxic biological reactions will degrade the substrate. Finally, fermentation or anaerobic biological reactions begin once oxygen and nitrate, the electron acceptors, are depleted.

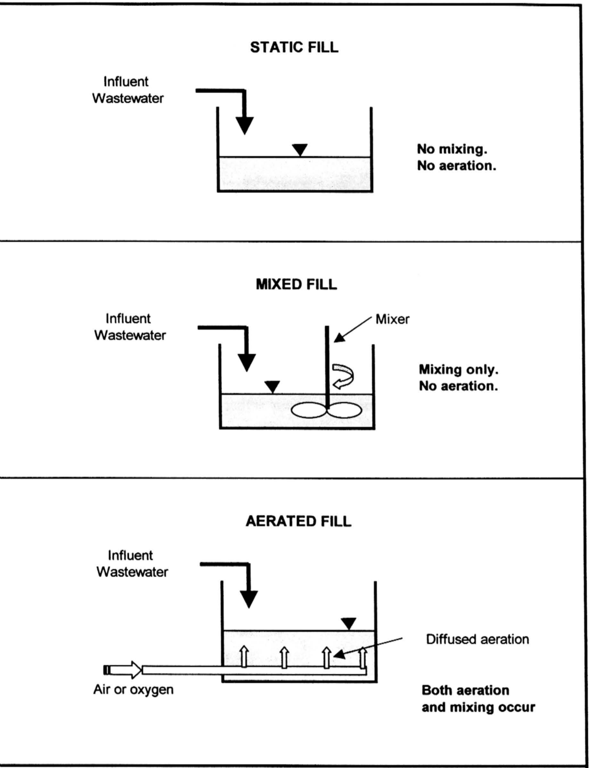

Figure 2.13 : Various Types of Fill for SBR Operation

Source: Ketchum, 1997

Aerated Fill is achieved by providing aeration during Fill. The rate of substrate degradation is limited either by the biological reaction rate which is a function of biomass and substrate concentrations when dissolved oxygen concentration is above some minimum concentration, or a function of the rate at which oxygen is supplied from aeration. In the first case, the SBR reactor size is smaller, but the aeration system is larger and energy input greater. In the second case, a larger reactor is required, but the aeration system is smaller and energy input less.

The number of SBRs in parallel is determined mainly by the design influent flow rates. Capital cost for single SBR system is lowest, because fewer components are required and the control system is simple. However, standby components or redundance capacity is expensive or impossible to provide. Single-tank SBR systems are possible when influent storage is provided upstream, when no influent flow occurs for a period of time sufficient to complete Settle and Draw.

Multiple-tank SBR systems are common for most municipal and continuous industrial influents and for systems where Fill is not allowed to overlap with Settle and Draw. In a two-tank SBR system, the time for Fill (t_f) in one SBR must equal the time for React, Settle, Draw, plus Idle (t_r, t_s, t_d, t_i, respectively) in the other SBR. Equation 1 shows this relationship for any number of SBRs designed for parallel operation, where n equals the number of tanks (equal-size) in parallel.

$$t_f * (n-1) = t_r + t_s + t_d + t_i \qquad \text{(Equation 1)}$$

The total liquid volume of the SBR tank (V_t) includes the volume occupied by the settled biomass and treated supernatant remaining at the end of Draw (V_0), and the volume added during Fill (V_f). Determination of the magnitude of V_0 will be described below in the section considering system design. The design influent flow rate (Q) multiplied by t_f gives V_f. These relationships are shown in Equations 2 and 3 below:

$$V_t = V_0 + V_f \qquad \text{(Equation 2)}$$

$$V_f = Q * t_f \qquad \text{(Equation 3)}$$

2.3.2 React Phase

During the React Phase, the reactor contents are mixed, with or without aeration. Mixed React results in denitrification, if organics and nitrates are both present. This denitrification results in reduction of oxygen demand which give rise to the anoxic or anaerobic conditions required for biological phosphorus removal. Aerated React results in completion of aerobic reactions. If aeration is extended, the bio-solids are reduced. The typical times for the React sequence is between 1.0 hour and 3.0 hours (Ketchum, 1997).

Table 2.7 illustrates bases of design for common operating policies to meet selected treatment objectives.

Table 2.7: Common Operating Policies

Treatment Objective	Fill Policies	React Policies	Reference
1. Organic carbon and SS reduction, minimum energy consumption or sludge production	Static, Mixed, then Aerated	Aerated	Irvine *et al.*, 1985
2. Organic carbon and SS reduction, and nitrification	Static, Mixed, then Aerated	Aerated	Ketchum *et al.*, 1979
3. Organic carbon and SS reduction , and denitrification	Static, Mixed, then Aerated	Aerated, followed by Mixed then Aerated	Alleman & Irvine, 1980
4. Organic carbon and SS reduction, and biological phosphorus reduction	Static, Mixed, then Aerated	Aerated	Ketchum *et al.*, 1987
5. Industrial organic wastewater, toxic at high concentration	Mixed (short period) then Aerated	Aerated (long period)	Asher *et al.*, 1992

Source: Ketchum, 1997

As indicated in **Table 2.7** above, once the treatment objective is determined the required operation and bases of design can be developed. As an example, this process will the described for treatment objective Number 4, organic carbon and suspended solids reduction and biological phosphorus reduction. As reported by Ketchum *et al.* (1987), cycle times that resulted in meeting this treatment objective in a two-tank systems were as follows: Static Fill – 2.30 hours; Mixed Fill – 1.00 h, Aerated Fill – 1.00 h (total Fill – 4.3 h); Aerated React – 2.00 h; Settle – 0.75 h; Draw – 0.75 h; and Idle – 0.8 h. This operating strategy was selected to enrich and support four important groups of organisms involved in SBR biological phosphorus reduction – denitrifying organisms, fermentation-product-manufacturing organisms, phosphorus-accumulating organisms, and aerobic autotrophs and heterotrophs. During Static Fill, the raw substrate concentration would increase as there is minimal contact between the settled biomass and influent substrate. At the beginning of Mixed Fill, oxygen would be present from the previous cycle, from atmospheric surface transfer, and from the influent wastewater. Oxidised nitrogen would also be present from the previous cycle and from the influent wastewater. During the early period of Mixed Fill, oxygen would be consumed by the heterotrophs in high substrate concentration. Anoxic conditions would then prevail as phosphorus-accumulating organisms compete with denitrifying organisms for substrate until the oxidized nitrogen is eliminated. The anaerobic conditions that then exist would favour fermentation-product-manufacturing organisms that use incoming raw substrate to produce biodegradable by-products such as acetic acid. At the same time, phosphorus-accumulating organisms would release stored polyphosphorus to provide energy needed to accumulate these by-products as high molecular weight intercellular fats. The released phosphorus would remain in solution. When aeration begins during Aerated React (and, in some situations at the end of Fill) aerobic conditions would develop to allow phosphorus-accumulating organisms to use stored intercellular fats for growth. During aerobic growth, the phosphorus-accumulating organisms would use the stored intercellular fats to provide energy needed to take up extracellular phosphorus and store it as intracellular polyphosphorus. Soluble phosphorus would be removed from solution and the phosphorus-accumulating organisms would be prepared to consume and store by-products during the next period of anaerobic conditions during Mixed Fill. Aerobic autotrophs and heterotrophs would then use residual substrate so that after Settle, a low-phosphorus, low-substrate treated supernatant exists (Ketchum *et al.*, 1987).

Treatment objective Number 1, organic carbon and suspended solids reduction, minimum energy consumption or sludge production, would be met by providing a brief period of Static and Mixed Fill, primarily to provide a short anoxic period. This has been found to select for organisms that settle better and to limit formation of filamentous organisms (Irvine *et al.,* 1985). The flexibility of the SBR allows operation to either minimize sludge production (i.e. by extending aeration during React, the system would behave in a similar way as a continuous flow Extended Aeration system) or minimize energy consumption (i.e. by extending Static and Mixed Fill, nitrate would become the electron acceptor thus reducing the need for aeration; and by limiting the period of Aerated React, more sludge would be produced and less aeration required).

Treatment objective Number 2, organic carbon and suspended solids reduction, and nitrification, would be accomplished by increasing the time of Aerated React to assure oxidation of the ammonia (Ketchum *et al.,* 1979). Treatment objective Number 3, organic carbon and suspended solids reduction, and denitrification would be accomplished similar to Number 2, except that a period of Mixed React would be introduced near the end of React and shortly before a short period of Aerated React. Mixed React would enable the removal of nitrates produced from ammonia nitrification at the end of Aerated Fill and during Aerated React. The short period of

Aerated React at the conclusion would be required to assure an oxidized treated effluent (Alleman and Irvine, 1980).

Treatment objective Number 5, industrial organic toxic at high concentration, would be met by long periods of aeration. This would limit the substrate concentration from reaching toxic levels. Most of the Fill cycle would be aerated to achieve this. However, a short period of Mixed Fill would be needed to avoid development of filamentous bacteria and poor settling sludge (Asher *et al.*, 1992).

2.3.3 Settle Phase

The Settle sequence is activated after the appropriate aeration or react time. Settling is based on subsidence of the sludge blanket layer and the concentration of the settled biomass. During the Settle Phase, MLSS is separated from the treated wastewater by quiescent settling (i.e. conditions are absolutely still during settling; apart from the settling of the solids, there is no other movement within the tank). During this phase, absolutely no mixing or agitation of any sort is carried out, thus settling occurs under truly quiescent conditions (Ketchum, 1997).

2.3.4 Decant Phase

The Decant Phase involves the withdrawal of treated wastewater from the reactor. This is the supernatant after settling of the reactor contents occurs.

2.3.5 Idle Phase

During the Idle Phase, waste sludge is removed from the reactor bottom. The idle cycle may be omitted by wasting sludge near the end of the react or decant cycle.

2.4 SYSTEM DESIGN

Municipal SBR systems require at least two parallel SBRs to avoid the need for Fill to coincide with Settle and Draw. As the number of tanks increases, the total tank volume required decreases for a fixed hydraulic retention time. This is because the sum of the time needed for Fill in all tanks equals the time for React, Settle, Draw, and Idle in one tank. Therefore a smaller fraction of the time is dedicated to Fill for each cycle when more tanks are used. However, the system requires more component parts (e.g. aerators, decanters and inlet valves) and becomes slightly more complicated to operate when more tanks are used (Ketchum, 1997).

2.4.1 Cycle Times

Once a hydraulic retention time and number of parallel units are selected the size of each tank and its components can be determined. Hydraulic retention times range from 6 hours to 24 hours. Total time required frequently results in about four cycles per day (Ketchum, 1997).

Settle (t_s) is based on subsidence of the sludge blanket layer (usually at a rate greater than one meter in 10 minutes), and the concentration of the settled biomass. Typical times range from 0.5 h for shallow tanks to 0.75 h for deep tanks, and 1.0 hour is often selected for conservative design. (Ketchum, 1997).

- Draw (t_d) is based on decanter hydraulic capacity, cost of decanter of effluent pump and concern for instantaneous high effluent flow rates. Typical times are 1.0 h or greater for large systems.
- React (t_r) is based on the treatment objectives, reaction rates, time during Fill dedicated to aeration and the need to provide Mixed React. Typical times are 1.0h to 3.0 h.
- Idle (t_i) is based on the need for flow equalization and uncertainty of flow rate fluctuations. Can be near zero for the peak design flows if the fluctuations are well defined. A longer design Idle time offers a convenient method of increasing other cycle times when future conditions change.
- Fill (t_f) is computed from Equation 1.

2.4.2 Tank Size

The SBR hydraulic retention time selected for design is used to determine the total tank volume, V_t. Equation 3 is used to determine the volume added (V_f) to each tank during the design Fill and Draw. The volume retained in each tank at the end of Draw and beginning of Fill, V_0, is determined by rearranging Equation 3, thus $V_0 = V_t - V_f$. Designers of SBRs favour deep tanks because aeration efficiency is improved with depth, a greater fraction of the supernatant can be removed during Draw, tank construction costs are less as compared to long, narrow and shallow tanks, and land area requirements are less. The minimum distance between the decanter and the sludge blanket layer surface will be suggested by the supplier of the equipment. Once these depths are determined, the depth associated with V_0 is compared to the depths associated with the decanter and the tank surface area. A larger volume can be decanted for typical domestic wastewaters in SBRs designed on the basis of hydraulic retention time. Domestic wastewaters generally do not require consideration for chemical addition because they contain abundant nutrients. (Ketchum, 1997).

2.4.3 Food per Mass (F/M) Ratio and Selective Pressures

In an SBR's cycle of operation, the oxygen uptake rate and F/M ratio are constantly changing. When aeration is initiated, after the "Anoxic Fill" period, the F/M ratio can range from 0.6 to 1.0, or higher, and the oxygen uptake rate can exceed 125 mg/l/hr. At the end of the aeration period the F/M ratio and uptake rate should be near zero. It is this non-steady state operation that is the key to the SBR process. The wide swings in the F/M ratio place *selective pressures* on the biomass. The period when the F/M ratio and uptake rate are high is a "feast" period; food and air is plentiful, and the D.O. level stays near zero because the oxygen demand exceeds the aeration system's maximum capacity. This "feast" period inhibits the growth of slow growing filaments, and encourages the growth of floc forming zoogleal organisms. The period when the F/M ratio and uptake rate are low is a "famine" period; all available food has been utilized. The "famine" period inhibits the growth of fast growing filaments, and again encourages the growth of desirable organisms. Design of an SBR should be based on aerobic F/M. (Ketchum, 1997; Irvine *et al.*, 1997).

2.5 SBR SYSTEM COMPONENTS

The principal components are the tank, inlet, outlet, mixing and aeration system, and controller. The tanks tend to be relatively deep, but width to length ratios are unimportant (Norcross, 1992). Municipal systems usually are designed using concrete tanks, but steel cylindrical tanks have on occasion been used with appropriate corrosion control.

2.5.1 Inlet

Influent is pumped directly into an individual SBR or through a manifold and automatic valves. The influent is introduced near the surface if a period of Static Fill is desired. Many systems introduce the influent into an inlet system or chamber designed to force the influent into the tank near the bottom, below the settled biomass, to provide some mixing. This may be accomplished with a simple inlet chamber baffled to force the influent in at the bottom, or a pipe distribution system is used to distribute more uniformly the influent throughout the settled biomass (Norcross, 1992).

In the case of the KLIA plant, the influent to the SBR tank is introduced via pipe near the base at one corner of the tank.

2.5.2 Decanter

A wide variety of decanter mechanisms have been used to remove the treated supernatant. Most are designed to float or move downward to withdraw the supernatant from slightly below the water surface, and others are fixed at an elevation to withdraw the supernatant from above the expected settled sludge blanket and below the water surface at the end of Draw (Norcross, 1992). Advantages of the first type (floating or movable type) include:

- the ability to begin Draw before the sludge reaches its lowest level,
- the ability to always Draw from a fixed distance below the water surface, and
- the added flexibility to easily change high water and low water levels, even from cycle to cycle.

The decanters used in early SBRs were, basically, pipes with drilled holes along the bottom or side. During the "React" and "Settle" periods solids would tend to get drawn into these early decanters, thus the effluent from the first half minute to one minute of "Decant" would contain solids. In the past several years, all SBR manufacturers have developed "solids excluding" decanters. These decanters either float in the basin at all times and are operated by hydraulic differential, or are electrically driven and remove the effluent from beneath the liquid surface, thus avoiding scum and foam present. A floating decanter allows reduced "Settle" periods since the "Decant" period can begin with the sludge blanket is approximately 1.2 meters below the surface level, while a fixed decanter requires that the sludge blanket be approximately 2.4 meters below the surface before the "Decant" period can begin. Fixed decanters may be appropriate in certain applications (Norcross, 1992).

The KLIA plant uses a moving-arm decanter. The maximum decant depth is 1m. The trough of the decanter, about 3m in length, is attached to the moving arm which is controlled by a motor. The rate of decanter descent can be increased or decreased as necessary. The maximum recommended loading rate is 20 m^3/m.h.

2.5.3 Aeration

Many SBR systems are designed to use fine bubble and coarse bubble aeration systems, while some manufacturers utilize jet-aeration systems. Jet aeration offers significant advantages in the SBR process when compared to either fine or coarse bubble diffusers. Unlike diffuser systems, a jet aerator can mix without aerating; making an anoxic mix period possible if the process requirements so dictate. (Norcross, 1992; Tchobanoglous and Burton, 1991).

The KLIA plant uses fine pore membrane diffusers, mounted on air manifold pipes which run the length of the tank close to the bottom. The diffusers are installed in a grid pattern on the bottom of the aeration tank to provide uniform aeration throughout the tank.

Automatic controllers may be simple, designed to operate only from a timer and liquid level switches. For example, in a simple two-tank system, Fill of Tank 1 begins when the high water level switch in Tank 2 indicates a need to open the inlet valve of Tank 1 and close the inlet valve of Tank 2. Mixed Fill begins at a fixed time later and Aerated Fill begins after that. To assure that adequate time is provided to complete aerobic reactions during peak flow periods, intermediate liquid level switches are used. These intermediate liquid level switches begin mixing or aeration sooner if the liquid level reaches a high level earlier than expected under normal conditions. Tank 2 begins Aerated React as soon as it reaches the high water level and Fill is discontinued. Aerated React continues for a fixed period of time, unless an intermediate water level is indicated in Tank 1 before it is expected during normal flow conditions. At the end of the Aerated React time, aeration and mixing is stopped. Settle is then allowed to occur for a fixed period of time. Following Settle, Draw is begun and continues until a selected low water level is indicated. Idle occurs until Fill is required. This type of control is favoured for small systems and for systems when influent flows and characteristics are reasonably predictable.

More complex control systems allow greater energy conservation, the use of smaller tanks, and more precise control. These systems use sludge blanket level indicators to begin Draw as soon as possible, and to stop if the sludge blanket settles to an unexpected high depth. This allows maximum supernatant removal and as easily as possible after Settle begins. Oxygen concentration indicators allow more efficient use of aeration. These systems are favoured for larger systems where more skilled operators are available, and savings in energy consumption and system size warrant the increased cost of the control system and operation. (Ketchum, 1997).

According to Norcross (1992), the best operating strategy uses a flow proportional strategy that matches aeration to the influent flow rate. A dedicated computer is used to implement the strategy. This strategy provides minimum aeration at minimum flows, and increases the aeration proportional to batch size. The computer will continuously calculate the influent flow rate by measuring the rate of change of level in the filling reactor. This allows the computer to calculate required aeration for batches of all sizes.

2.6 EFFICIENCY OF SBR PROCESS

The full scale results shown below have proven the efficiency of the SBR process. (Note: only results of municipal SBR plants shown here). The results shown in **Table 2.8**, indicate that the SBR plants achieved more than 90% reduction of BOD and SS, and reduction was usually higher than 97%. The plants at Oak Pointe, Flushing and Lake Edgewood also reported reductions in ammoniacal nitrogen (more than 90% reduction) and phosphorus (ranging from 88% to 98% reduction).

Table 2.8: Removal Efficiency of Some Municipal SBR Plants

Location	Influent	Effluent	Removal Efficiency
Harrah, Oklahoma 0.25 MGD	191 mg/l BOD 157 mg/l SS	3 mg/l BOD 4 mg/l SS	98.4% 97.5%
Del City, Oklahoma 2.59 MGD	172 mg/l BOD 171 mg/l SS	5 mg/l BOD 7 mg/l SS	97.1% 95.9%
Oak Pointe, Michigan 0.057 MGD	164 mg/l BOD 139 mg/l SS 21 mg/l NH3 6.8 mg/l P	2 mg/l BOD 9 mg/l SS 0.7 mg/l NH3 1.7 mg/l P	98.8% 93.5% 96.7% 75.0%
Flushing, Michigan 1.76 MGD	114 mg/l BOD 115 mg/l SS 11.7 mg/l NH3 2.6 mg/l P	3 mg/l BOD 8 mg/l SS 0.8 mg/l NH3 0.3 mg/l P	97.4% 93.0% 93.2% 88.5%
Kearney, Montana 0.34 MGD	383 mg/l BOD 481 mg/l SS	9 mg/l BOD 10 mg/l SS	97.7% 97.9%
Lake Edgewood, Michigan 0.028 MGD	295 mg/l BOD 663 mg/l SS 23 mg/l NH3 14 mg/l P	5 mg/l BOD 14 mg/l SS 0.9 mg/l NH3 0.9 mg/l P	98.3% 97.9% 96.1% 93.6%

Source: Norcross, 1992

In a study carried out by Rim *et al.* (1997), a full-scale pilot plant using SBR process was experimented, using wastewater generated from a recreational centre. The main purpose of this investigation was to evaluate applicability in the field and process removal efficiencies in terms of BOD, SS, TN and TP and its corresponding kinetic parameters. It was found that removal efficiencies were comparable with results of other studies reported in literature. BOD removal was observed to be 95% on average while SS removal to be 89% on average. Removal rate of nitrogen was 70% in terms of total nitrogen and that of phosphorus was 77% in terms of total phosphorus. Effluent concentrations were 7.3 mg/l BOD, 10.4 mg/l SS, 13.6 mg/l TN, and 0.9 mg/l TP. Effluent quality was maintained consistently stable by controlling decantation quantity and operating cycles. Denitrification/nitrification were accomplished during anaerobic/aerobic processes.

The SBR was operated 2 to 4 cycles per day (6 to 12 hours per cycle). Operating condition for the experiment is shown in **Table 2.9** and its operating strategies is shown in **Table 2.10. Table 2.11** shows the concentration of influent and effluent for this facility

Table 2.9: The Operational Conditions

Cycle time	6 hours	12 hours	8 hours	6 hours
Operating Periods	Day 0-180	Day 181-270	Day 271-330	Day 331-400

Source: Rim *et al.*, 1997

Table 2.10: Time schedule mode during a cycle

Cycle mode	Time (minutes)					
	Fill	Anaerobic mix	Aerobic mix	Settle	Draw	Idle
6 hours/cycle	20	160	280	340	350	360
8 hours/cycle	20	240	410	460	470	480
12hours/cycle	20	360	600	660	690	720

Source: Rim *et al.*, 1997

Table 2.11: Influent and Effluent Characteristics of Sewage

Parameter	Influent			Effluent			Average Removal Efficiency
	High	Low	Average	High	Low	Average	
BOD (mg/l)	225	76	139	18.4	1.0	7.3	94.7%
SS (mg/l)	180	11	72	44	1.0	10.4	85.6%
TN (mg/l)	75.1	25.3	45.0	38.5	10.8	0.1	69.8%
TP (mg/l)	8.8	1.1	3.9	7.3	10.4	13.6	76.9%

Source: Rim *et al.*, 1997

Pollutant removal rate by varying the cycle time is shown in **Table 2.12.** BOD was not influenced by change of cycle. SS concentrations got worse at 12 hours/cycle, pin floc occurred due to destruction of floc of activated sludge by long period of aeration. Ammoniacal nitrogen removal rate was more than 90%, and was not dependant on the change of cycle. Removal rate of total phosphorus was improved by prolonged operating time (12 hours/cycle).

Table 2.12: The effects of operating cycle time on removal efficiency

Parameter	**Removal efficiency**		
	6 hours/cycle	**8 hours/cycle**	**12 hours/cycle**
BOD	95.2%	94.2%	93.3%
SS	80.2%	84.5%	76.4%
TN	62.8%	65.1%	55.7%
NH_4-N	90.5%	90.1%	92.1%
TP	67.3%	72.7%	75.8%

Source: Rim *et al.*, 1997

BOD removal rate was always over 90% in spite of different operating conditions. 70% of the BOD was removed during anaerobic conditions and 25% of the remainder was removed during aerobic condition.

In the case of TN, denitrification was completed after 1.5 to 2.0 hours during anaerobic conditions. Nitrogen removal rate ranged between 50% - 90% and average removal rate was around 70% with heavy fluctuations. **Figure 2.14** shows a variation of nitrogen concentration for 8-hour cycle. Ammoniacal nitrogen was constant under anaerobic conditions, on the other hand, it declined to almost zero at the end of aerobic time with nitrification. Nitrate nitrogen was almost denitrified in early anaerobic conditions and more than 90% was removed in 1 hour.

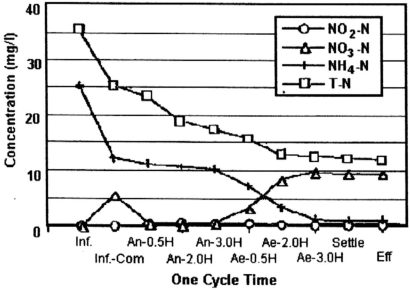

Figure 2.14: Variation of Nitrogen concentration in the bulk solution for 8-hour cycle

Source: Rim *et al.*, 1997

Phosphorus removal rate was relatively good for the longer cycle times. Heavy fluctuation of removal rate was influenced by the BOD/TP ratio in the influent and the SS concentration in the effluent. Most of the TP in the treated water existed as PO_4-P and the rest were released from SS. **Figure 2.15** shows a variation of concentration for the 8-hour cycle. It shows that phosphorus is released during anaerobic conditions, and phosphorus luxury uptake during aerobic conditions occurred.

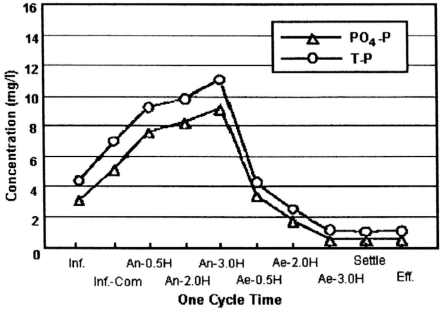

Figure 2.15: Variation of Phosphorus concentration in the bulk solution for 8-hour cycle

Source: Rim *et al.*, 1997

2.7 SBR IN TROPICAL CONDITIONS

While most of the research in SBR has been carried out in temperate conditions, the SBR has also been applied in tropical countries around the world. The results recorded thus far indicate that SBR has been used successfully in treating both domestic wastewater and industrial wastewater in tropical climates (Andreadakis *et al,* 1995; Gharagozian and Samstag, 1998; Keller *et al,* 1997; Lansdell, 1996; Zhang *et al,* 1996).

A study of the sewage pollution control in the Wider Caribbean Region (Gharagozian and Samstag, 1998), identified several wastewater treatment technologies and water quality standards for this region. The SBR was accepted as one of the wastewater treatment technologies appropriate for this region (simple to operate and maintain, able to reduce target pollutants such as BOD, SS, and nutrients to acceptable levels). Two of the plants surveyed in Venezuela used a modified SBR for treating the municipal wastewater. They were in La Mariposa (population 770,000) and Juangriego (population 50,000). The plants have consistently produced effluent in the range of 5 to 15 mg/l of SS and BOD. The report also recommended domestic and industrial wastewater discharge standard for sensitive waters. Some of the parameters for which these discharge standards were recommended are: BOD – 30 mg/l, SS – 30 mg/l, COD – 150 mg/l, Ammoniacal Nitrogen – 5 mg/l.

In the treatment of septage originating from cesspools serving non-sewered areas in the Caribbean, Andreadakis et al (1995) found that the SBR was able to treat this type of load quite well. Septage is characterised by high strength (2.5 times stronger than typical domestic sewage). Flocs from septage settle well and allow for a high MLSS concentration (up to 8,000 mg/l). A single-stage batch aerated system with a reactor volume of 1.6 times daily septage flow, a solids retention time (SRT) of 15 days and COD loading of 0.15 mg/mg MLSS/day produced a well-stabilised sludge and good nitrification (the average nitrogen removal rate was to the order of 70%. The anoxic second-stage post denitrification resulted in an overall nitrogen removal of 88%. Further improvement of the system, with nitrogen removal of about 95% and average effluent nitrogen concentrations lower than 10 mg/l could be achieved by adoption of a two stage system consisting of a first aerated stage, followed by a second stage alternating aerated and anoxic cycles, and addition of external carbon during the anoxic cycle.

In treating industrial wastewater, additional challenges include the higher concentrations of both carbon and nutrients in the raw wastewater. Research conducted by Keller *et al* (1997), in Queensland, Australia, addressed the issue of abattoir effluent. A single tank SBR was used. The results from the laboratory work were excellent but are yet to be confirmed by pilot plant studies. It was found that a certain amount of anaerobic pre-treatment could reduce part of the carbon content, while still leaving sufficient COD required for successful Biological Nutrient Removal. Influent concentrations were high, approximately 190 mg/l Nitrogen and 50 mg/l total

Phosphorus, yet effluent quality of less than 20 mg/l N and less than 5 mg/l total P were achieved.

In Hawaii, Zhang et al (1996) evaluated the performance of an Anaerobic SBR for treatment of swine waste with short hydraulic detention time (two to six days) and found that volatile solids reduction ranged from 39 to 61 percent while the BOD reduction ranged from 58 to 86 percent. Surprisingly, a better removal was achieved with a three-day detention time than with a six-day detention time.